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The Economics of Dead Zones: Linking Externalities
from the Land to their Consequences in the Sea

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Abstract: The purpose of this review and analysis is to provide a basic understanding of the issues related to worldwide hypoxic zones and the range of economic questions sorely in need of answers. We begin by describing the causes and extent of hypoxic zones worldwide, followed by a review of the evidence concerning ecological effects of the condition and impacts on ecosystem services. We describe what is known about abatement options and cost effective policy design before turning to an analysis of the large, seasonally recurring hypoxic zone in the Gulf of Mexico. We advance the understanding of this major ecological issue by estimating the relationship between pollutants (nutrients) and the areal extent of the hypoxic zone. This "production function" relationship suggests that both instantaneous and legacy contributions of nutrients contribute to annual predictions of the size of the zone, highlighting concerns that ecologists have raised about lags in the recovery of the system and affirms the importance of multiple nutrients as target pollutants. We conclude with a discussion of critical research needs to provide input to policy formation.

The Economics of Dead Zones: Linking Externalities from the Land to their Consequences in the Sea

Introduction

The BP Deepwater Horizon oil spill in the Gulf of Mexico was a major environmental news story throughout the spring and summer of 2010. The daily headlines expressed concern about long term damage to ecosystems and wildlife habitat. Casual readers could easily have been led to believe that the spill and its aftermath represented the most significant and enduring environmental threat to this region. However, equally challenging threats to the environment exist in this region and chief among them is the seasonal hypoxic or "dead" zone that occurs annually off the coast of Louisiana and Texas. What may come as even more of a surprise to those not familiar with hypoxic zones in freshwater and coastal ecosystems is that such dead zones have been appearing worldwide at proliferating rates (Diaz and Rosenberg, 2008). Nutrient over-enrichment is the main reason why these zones form and nutrient-fed hypoxia is now ranked as an important threat to the health of aquatic ecosystems, including oceans (Rockström et al. 2009).

The rather alarming term "dead zone" is surprisingly appropriate: hypoxic regions exhibit oxygen levels that are too low to support aquatic organisms that would otherwise be found there, including commercially desirable species. While some are naturally occurring, the number, size and duration of these zones have been significantly expanded as a result of the anthropocentric contributions of nutrients coming from mostly agricultural, municipal, and industrial sources.

There is a fertile ground for economists to plow as they explore the consequences of the stressors arising from human alteration of ecosystems. An abundance of the knottiest issues studied in our field are exhibited in the 'externality' problems related to hypoxic conditions: uncertainty and potential irreversibility, spatial interdependence of individual agent decision

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making and associated externality damages, transboundary pollution problems, valuation of poorly understood ecosystem services, and nonpoint source pollution generated over vast agricultural landscapes with their associated measurement and policy design problems. We begin introducing what is known about the causes, duration, and location of the major hypoxic zones worldwide. Next, we discuss the economic issues associated with this environmental externality, beginning with what is known about the changes in ecosystem services likely to occur because of the widespread presence of hypoxic conditions. In the third section, we briefly describe the conceptual issues involved in analyzing large-scale marine eutrophication problems and describe a limited number of existing studies related to the costs of reducing hypoxic conditions and the type of analysis needed to better understand the policy options. In the fourth section, we return to a more in-depth analysis of a specific hypoxic zone: the annually recurring zone in the northern Gulf of Mexico where we advance the understanding of this major ecological issue by econometrically estimating the production relationship between the primary pollutants and the size of the hypoxic zone. This "production function" relationship provides an example of useful cross-fertilization of ideas from basic economic analysis to ecology and suggests that both instantaneous and legacy contributions of nutrients contribute to annual predictions of the size of the zone, highlighting concerns that ecologists have raised about multiple nutrients and lags in the recovery of the system. We use the estimated relationship to comment on the existing policy goals for the Gulf of Mexico hypoxia. We conclude by discussing further research that is needed to provide input to policy formation.

I. An Overview of the Causes and Consequences of the Hypoxic Zone

The term "hypoxia" simply means a deficiency of oxygen and is operationally defined by convention as water with less than 2 mg/L (anoxia means that there is no oxygen). Hypoxic

zones in both coastal oceans and freshwater systems have occurred naturally in areas that have the requisite combination of weather patterns, ocean geography, currents, and nutrients, but their magnitude and extent around the world have increased dramatically over the past 50 years as a result of human activities (Diaz and Rosenberg 1995; Diaz and Rosenberg 2008). Prior to 1970, "scattered reports" of coastal hypoxic zones in Europe and North America appeared in the literature (Rabalais, Diaz et al. 2010) but, such reports were more common by the 1990s, and by 1995, Diaz and Rosenberg (1995) reported over 195 documented cases worldwide. This number had doubled to just over 400 zones by the time they completed their 2008 review, to which an additional 115 sites in the Baltic Sea have been added (Conley, Carstensen et al. 2011). Furthermore, there is evidence that many of these locations were not hypoxic when data first became available, thus the rapid increases is not ascribable to increased scrutiny of the condition (Rabalais, Diaz et al. 2010).

The map in Figure 1 is based on the Diaz and Rosenberg census, and identifies the location of documented hypoxic zones around the world. Casual inspection highlights their contention that coastal hypoxic zones are primarily located downstream of major population centers and/or land impacted by agriculture.

The behavior of many animals that live in coastal zones is observably altered when hypoxia occurs, but the response is not homogenous. Researchers increasingly caution that a single threshold value is unlikely to be an accurate marker for all hypoxic conditions and affected species (Vaquer-Sunyer and Duarte 2008; Steckbauer, Duarte et al. 2011)[1]. When levels drop below 0.5 to 1 mg/L, large-scale mortality may result if organisms are not able to flee the area. The duration of hypoxic conditions is also a factor in mortality events.

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[1] A review of definitions and values is provided in Rabalais, Diaz et al. (2010).

About half of the hypoxic zones around the world are seasonal – oxygen depletion occurs in spring and summer after phytoplankton growth increases as a result of nutrient enrichment (Diaz and Rosenberg 2008). The typical length of a seasonal hypoxic zone is from a few weeks to several months, but these events can range from just a few days to much longer time periods. In some locations (about 8 percent worldwide), hypoxic conditions occur continuously throughout the year. Likewise, the size of annually recurring zones can vary significantly from year to year. For example, the size of the Gulf of Mexico's hypoxic zone has ranged from a low of 40 km^2 in the drought year of 1988 to over 20,000 km^2 in both 2001 and 2008, since measurements were initiated in 1985 (Figure 2). The anthropogenic causes of hypoxia can be linked to rapidly increased use of nutrients (primarily nitrogen and phosphorus) in agricultural production, waste water from human populations and industrial sources, and the burning of fossil fuels. For example, there is three times as much reactive nitrogen and phosphorus being dispersed into the world's oceans today compared to pre-industrial times (Bennett, Carpenter et al., 2001; Galloway, Townsend et al. 2008; Diaz, Rosenberg et al. 2009; Rabalais, Diaz et al. 2010; Turner, Rabalais et al. 2012). While nutrients are valuable inputs to agricultural production, they become externalities when they are introduced to local rivers and streams via runoff from farm fields, urban landscapes, or from municipal wastewater, and when they are transported further downstream to larger freshwater lakes, to estuaries, to coastal ecosystems, and the open ocean. Once deposited at these locations, the nutrients feed the growth of phytoplankton, creating at times noxious blooms of plant growth. When this plant material dies, it falls naturally through the water column and decomposes in the bottom water layer and sediments. The process of decomposition results in reduced oxygen levels if the consumption of oxygen exceeds the re-aeration of oxygen in a stratified physical system. The increased rate of

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carbon production is referred to as "eutrophication," which can give rise to a number of negative effects on ecosystems such as the noxious and toxic algal blooms, oxygen deficiency, unpleasant odors, habitat alterations that negatively affect living resources and nutrient and food cycles, and cause the degeneration of both aesthetic and economic values (Rabalais 2004). The largely unseen depletion of oxygen in the water is considered one of the most serious ecological consequences of unacceptable environmental change (Diaz and Rosenberg 2008).

Nutrients are essential to plant growth, both terrestrial and aquatic, and support increased productivity in agricultural crops and coastal and ocean fisheries. However, the many-fold increase in nitrogen and phosphorus reaching aquatic systems is problematic. Nitrogen and phosphorous both contribute to the problem, but plants respond to these inputs in fixed factors; i.e., they exhibit a Leontief style production function. In some hypoxic zones, it is clear that nitrogen is the limiting nutrient, in others, phosphorous, and yet others appear to be heterogeneous (Savchuk and Wulff, 2009; Quigg et al., 2011). Often, there is an active debate concerning which nutrient is acting as the driver in creating eutrophication and hypoxic conditions (Conley, Paerl et al. 2009).

A notable case of anthropogenically magnified hypoxia occurred in the northwestern shelf of the Black Sea adjacent to the outflows of the Danube and Dnieper rivers. Its notoriety has much to do with the system's partial recovery from the condition since anthropogenic nutrient inputs dropped rapidly and precipitously with the breakup of the Soviet Union and Eastern Europe's economic transition (Mee 2005; Levin, Ekau et al. 2009). Furthermore, this natural experiment provided a unique opportunity for researchers to observe the path of ecosystem response first to increased nutrient loads and then to the subsequent rapid. A brief review of the Black Sea case serves several purposes. It allows us to describe a common pathway

through which anthropogenically-caused hypoxic conditions become problematic, it demonstrates that these conditions are potentially reversible, and it also serves to highlight the fact that large natural systems, just like economic systems, are often impacted by multiple shocks which can exacerbate nonlinear response and alter the path towards long run equilibriums.

Agricultural (crop and animal production) and industrial activity expanded rapidly around the Black Sea in the 1960s resulting in rapid influxes of nutrients which resulted in the eutrophication of the system; there were harmful algal blooms, fish kills, and an expanding hypoxic zone (Mee 2005; Selman 2008). A dead zone of 3,500 km^2 was observed in summer 1973. The zone increased nearly tenfold by 1978 and hit a peak size in the late 1980s when it measured 40,000 km.^{2} Pictures of rotting sea life that ended up on Romanian and Ukrainian beaches became common. During this period, the fishery that once supported over 25 species of commercially valuable fish and was worth roughly $2 billion dollars was reduced by 90 percent. Other economic impacts included an estimated $500 million in lost tourism revenue (Battaglini 2008). While studies assessing the nonmarket impacts are not available, it is likely that the health effects, (more than 20,000 instances of serious waterborne illness were reported (Battaglini 2008)), lost recreation opportunities, biodiversity losses, aesthetic concerns, and other ecosystem losses were significant. To complicate matters, overfishing was occurring throughout the 1970s and 1980s, and an alien jellyfish (*Mnemiopsis leidyi*) was introduced into the ecosystem which, possibly aided by hypoxic conditions, outcompeted many native species. In short, very few of the signature attributes of this once diverse and productive ecosystem remained by the early 1990s (Zaitsev 1992).

A dramatic natural experiment then occurred in response to the collapse of the Soviet Union and resulting restructuring of the economies adjoining the Black Sea in the early 1990s.

These events resulted in the abandonment of substantial agricultural production in the region, reduced or no fertilizer usage, and rapid diminishment of the large area of hypoxia on the northwestern shelf. Almost no discernible hypoxic zone remained by 1995, although a temporary setback occurred in 2001 when a late rainfall and high temperatures triggered a hypoxic event (Mee, Frederick et al., 2005).

II. Consequences of Hypoxic Conditions for Ecosystem Services

Scientific understanding is gradually increasing regarding the ramifications of growing nutrient pollution losses, to both coastal and freshwater systems, on ecosystem services and the resulting welfare changes; however, much still remains unknown. Barbier (2012) provides a recent overview of challenges in identifying and valuing ecosystem services in the coastal zones. As described in the case of the Black Sea, compelling evidence exists concerning the substantive negative effects that the delivery of nutrients created in valuable ecosystem services; however, a similar sequence is less clear in other cases. Additionally, there are some examples of beneficial effects, at least in the short run, of nutrient delivery or hypoxia or both. Ideally, effects on the full suite of ecosystem services as identified in the Millenium Ecosystem Assessment (2005) associated with hypoxic conditions would be identified and described (supporting, provisioning, regulating, and cultural services). However, the focus of most assessments to date has been on the effects of nutrient loading and hypoxia on a range of individual species and the overall impacts on commercial and recreational fisheries. Furthermore, a full accounting of the welfare effects of nutrient loading requires consideration of the effects at each step between the initial loading of nutrients to waterways (often far upstream of the ultimate zone location), to the eutrophication in both freshwater systems as nutrients traverse downstream and in the coastal

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waters where eutrophication may create dead zones. In this brief summary, we focus on the lost ecosystem services that occur downstream, i.e., in the vicinity of the dead zones themselves, but the reader should keep in mind that the externality damages from nutrient loading can begin at the point of origin of the nutrients as well as throughout their journey down stream. In fact, a number of authors have argued that for many dead zones such as the Gulf of Mexico, the upstream water quality impacts exhibit higher welfare losses than the downstream impacts (SAB 2007) and should be the primary driver in policy design.

We further divide our discussion into two components: 1) documentation of the effects of oxygen depletion on ecosystem services associated with aquatic ecosystems and, 2) the nascent economics literature focused on the welfare effects of these changes.

Positive and Negative Impacts of Nutrient Over-Enrichment on Aquatic Ecosystems

Nutrients are essential to plant growth, both terrestrial and aquatic, and support increased productivity in agricultural crops, submerged aquatic vegetation, and phytoplankton and food webs that support coastal and ocean fisheries. In many cases in the past fifty years, however, the addition of nitrogen and phosphorus at double- to triple-fold levels is leading to increasingly more negative impacts on ecosystems. During this time there has been an increase in human population, nitrogen and phosphorus fertilizer usage, fishing fleet capacity, and marine fisheries landings (Caddy 2001). As pointed out by many authors (Caddy 2001, Chesney and Baltz 2001; Breitburg, Craig et al. 2009; Breitburg, Hondorp et al. 2009), it is difficult to tease apart the effects of nutrient additions, changes in fisheries methods and intensity, altered habitat, altered hydrology, natural variability, and additional human-influenced changes in ecosystems from conditions of variably severe hypoxia on specific ecosystem effects.

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For example, Caddy (1993) found increasing fishery production and yield with increasing nutrient loading up to a point where eutrophication increased, resulting in increased seasonal hypoxia and permanent anoxia. Additional effects of increased nutrient loading were shifts in trophic structure (Turner, Qureshi et al. 1998; Turner 2001) and shifts in relative abundance of pelagic (greater tonnage but less value) fish landings compared to demersal fish and shellfish (decreased tonnage but more valuable) landings (Caddy 1993; Chesney and Baltz 2011). Likewise, what may appear to be beneficial effects of hypoxia may only be manifested in the short-term but not with regard to longer-term productivity or ecosystem function. For example, hypoxia in the Chesapeake Bay resulted in stressed bivalves which were subsequently eaten by bottom feeding fishes and crabs that ventured into the hypoxic water temporarily to feed. While this food source may benefit demersal feeders, the loss in productivity and function of bivalve populations is not known.

Such a situation of temporarily increased food resources has not been demonstrated in the northern Gulf of Mexico hypoxic area, where demersal fishes, shrimp and crabs are devoid over large bottom areas when the dissolved oxygen levels fall below 2 mg/L (Renaud 1996a,b and many subsequent authors). The presence of a large hypoxic water mass off the coast of Louisiana in mid-summer may concentrate brown shrimp into shallower coastal waters (Craig, Crowder et al. 2005; Craig 2011) making them more susceptible to predators, including humans with trawls resulting in increased catches, but the overall productivity of the brown shrimp population is diminished by the removal of these smaller shrimp from further increase in size before capture in farther offshore areas later in the season. The phenomenon known as a "jubilee" that brings low oxygen waters inshore and forces fish, shrimp and crabs to migrate into intertidal areas where

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they are more easily caught, may as likely result in a massive fish kill if the low oxygen conditions trap the sea life longer than they can survive.

Extensive literature documenting the impact of hypoxic conditions on organisms, populations and communities, and ecosystem function is summarized in several review articles (Diaz and Rosenberg, 2005, 2008; Levin, Ekau et al. 2009; Rabalais, Diaz et al. 2010) and a book linking multiple taxa and ecosystem responses (Levin, Ekau et al. 2009; Rabalais, Diaz et al. 2010). Individual organisms are subject to a range of behavioral and physiological impacts, reduced fitness or reproductive capacity, or death, resulting from increased predation, low oxygen, and often toxic hydrogen sulfide in anoxic bottom waters. Many mobile organisms, i.e., fish, shrimp, and crabs, will move from hypoxic zones as oxygen falls below a threshold, which is 2 mg/L in the northern Gulf of Mexico (Rabalais, Harper et al. 2001). Blue crabs in crab traps will die if covered with low oxygen waters as the hypoxic area moves shoreward with tidal cycles. Studies of the reproductive capacity of Atlantic croaker in hypoxic waters or concentrated on the edges of hypoxic zones in the Gulf of Mexico manifest in impaired reproductive structures, egg and sperm production, and hormone imbalances (Thomas and Rahman 2010, 2011). The results suggest severe reproductive impairment can occur over large coastal regions in marine fish populations exposed to seasonal hypoxia, with potential long-term impacts on population abundance. Other mobile demersal organisms, such as eels, smaller portunid crabs, and rock shrimp, are often seen swimming in the water column or at the surface when the lower part of the water column is hypoxic; these organisms are subject to increased predation. Benthic organisms begin to demonstrate visible stress by emerging from burrows, lying listless on the sediment surface, and eventually dying when the oxygen concentration falls low enough for long enough (Rabalais, Smith et al. 2001). The existence of these organisms indicates that fish are not

entering the area to feed. The benthic communities in seasonally severe hypoxic areas are less diverse (even across taxa), less abundant and compose less biomass than other benthic habitats with well-oxygenated waters. There are some benthic organisms (mostly annelids) that can survive reduced oxygen levels because of facultative anaerobic respiration and morphological features, such as large and numerous branchiae; others can delay recruitment into hypoxic bottom waters. The food resources available to demersal feeders during hypoxia or that relocate to once-hypoxic areas are limited during and after hypoxia (Baustian, Rabalais et al. 2009; Baustian and Rabalais 2009).

The impact of hypoxic conditions on the harvest rates of commercial important and recreational fisheries is sometimes not discernible (Baustian, Craig et al. 2009), but this may be due to the inability to detect short-term effects from annual data (Thomas, Rahman et al. 2007) rather than the beneficial or negative effects outweighing the other. Several studies have shown declines in abundance or catch per unit effort in populations from the Baltic Sea cod fishery in part due to decreased egg survival in low dissolved oxygen waters, Lake Erie, and the Gulf of Mexico (Vanderploeg et al. 2009; Downing et al. 1999; Zimmerman and Nance 2001; O'Connor and Whitall, 2007). An additional complication is that many fisheries have experienced heavy fishing pressure at the same time that hypoxic zones have been growing, making separate identification difficult (Bretiburg et al., 2009).

For a concrete example of the difficulties in identifying the effects of nutrient externalities on an individual species, consider the effect of hypoxia on penaeid shrimp reproductive cycle and distribution along the Louisiana shelf. The presence, severity, and extent of hypoxia are maximal in June through August. White shrimp (*Litopenaeus setiferus*) spawn in exactly the same habitat and time for hypoxia. It would be expected that this would have a

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detrimental effect on the population of white shrimp, but neither Zimmerman and Nance (2001) nor O'Connor and Whitall (2011) were able to detect a loss of yield for this species. A potential reason is that white shrimp inhabit shallower coastal and estuarine waters away from hypoxia. On the other hand, brown shrimp (*Farfantepenaeus aztecus*) spawn in deeper offshore waters when hypoxia is not widely distributed along the shelf. The postlarval brown shrimp migrate into wetland nursery habitat in late winter and spring when hypoxia is not usually present and then emigrate offshore as juveniles at the time and place where severe hypoxia occurs on the shelf. Hypoxia is likely to block the movement of juvenile brown shrimp into farther offshore waters concentrating them in nearshore waters where human predators with trawls can efficiently capture them. As a result these juveniles and smaller shrimp are not able to grow to a larger size and generate a greater secondary production in the offshore waters. Thus, the main effect of hypoxia is not mortality but loss of future production. Brown shrimp are also more affected by low dissolved oxygen water as they were found to avoid waters with dissolved oxygen concentrations < 2 mg/L while white shrimp avoid concentrations < 1.5 mg/L. Effects have been documented on brown shrimp distribution as related to low dissolved oxygen as well as catch per unit effort (Zimmerman and Nance 2001; O'Connor and Whitall 2011).

Much remains to be learned about the impacts of excess nutrient enrichment on aquatic ecosystems. But as this brief review highlights, the effects depend on a wide variety of factors including precipitation, fishing pressure and management policies, and whether the estuary is influenced by significant ocean currents and processes (such as the Gulf of Mexico) or whether it is sheltered from such impacts (such as the Chesapeake Bay and the Baltic Sea). For some species of interest, anthropogenic nutrient contributions can lead to short run increases in abundance; in other cases, the data clearly suggest a reduction in populations even in the short

run. In other cases, the data available do not provide clear evidence one way or another. In the longer run, even more uncertainty pervades the relationship between low oxygen concentrations, reproduction and success of species. There is clear evidence that species are stressed by continued low oxygen conditions and that there are broader effects on food chains and habitat availability; the long-term effects of these stressors on the ecosystem's ability to provide the full suite of ecosystem services (provisioning, regulating, supporting, and cultural) remains unclear. Despite the many uncertainties, some research identifying the short run economic effects of low oxygen concentrations has been conducted and we turn to a review of that literature now.

Welfare Effects of Hypoxia on Fisheries and Ecosystem Services

Economists have studied the effects that nutrient over-enrichment has had on both marketed goods (primarily commercial fisheries) and nonmarket goods (primarily recreational fisheries), though a great many aquatic ecosystem services have yet to receive any attention. Nonetheless, the studies to date provide valuable information about some of the economic impacts of hypoxic conditions and are quite constructive in identifying the challenges in assessing lost consumer and producer surplus from nutrient over-enrichment.

At least two studies consider economic benefits from reducing nitrogen loading and hypoxic conditions in the Baltic Sea. Turner et al. (1999) describe the multiple market and policy failures contributing to over-enrichment problems of the Baltic, including air and water pollution externalities, ground water depletion, over fishing, pool land-use policies, and market price interventions. Benefits are monetized via the use of stated preference studies that estimated the benefits associated with reduced nitrogen concentrations in the Baltic. A Swedish study presented respondents with a referendum style contingent valuation questions concerning their

willingness to pay for an action plan that would reduce the level of eutrophication in the Baltic sea to a sustainable level. An identical survey instrument was administered to a sample of Polish adults. The average willingness to pay for the described improvement was larger in Sweden by a factor of seven. Using these estimates as a basis to extrapolate to the relevant populations, Turner et al. compute that the total economic benefits to restoring eutrophication levels in the Baltic to the described level was $70 billion SEK per year (or about $10 billion USD). The authors emphasize the uncertainty of this figure, but its magnitude makes clear that large values are possible.

Gren (2001) used the same stated preferences estimates to evaluate the benefits and costs from four different policies to reduce nitrogen loading into the Baltic based on the degree to which contributing nations coordinate their actions. In addition to providing valuable information on the distribution of benefits across affected countries, Gren demonstrates the importance of coordinated actions in maximizing net benefits.

In the first of a series of studies of economic losses from low oxygen conditions in the Neuse River and Pamlico Sound estuary in North Carolina, Huang et al. (2010) found that hypoxic conditions were accountable for about 13% of the decline in brown shrimp landing in the early 2000s. Following this work, Huang and Smith (2011) fit a bioeconomic model for brown shrimp and evaluate the optimal harvest path in the presence of hypoxia. They found that the optimal harvest profile occurs earlier in the presence of hypoxia than in its absence. By combining the harvest functions with a logit model describing shrimpers participation decision and demand for shrimp, Huang et al. (2012) consider the producer and consumer surplus losses from hypoxia. Given the small market share of shrimp coming from this fishery, they found that the demand curve is essentially flat so that all lost surplus accrues to producers.

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In a study of the blue crab fishery in North Carolina, Smith (2007) developed a bioeconomic model to evaluate consumer and producer losses from reduced hypoxic conditions. He incorporated key behavioral and biophysical components necessary to capture the key dynamics in the system including the relationship between hypoxia and population dynamics, spatial aspects where hypoxic conditions occur in one patch and not in another, and the importance that the management regime plays (i.e., open access vs. effort limits). He found that the total benefits from reduced hypoxic conditions (consumer plus producer surplus) are not large, ranging from $1 to $7 million annually, but that the size of those benefits depend importantly on management regime.

The effects on commercial and recreational fisheries in the Chesapeake Bay have also been studied. Massey, Newbold, and Gentner (2006) linked a bioeconomic model of Atlantic Flounder to a recreation demand model to evaluate the welfare effects of changes in oxygen concentrations off the Atlantic Coast to this important saltwater recreational fishery. Their bioeconomic model incorporates both the short run effects on species crowding and abundance discussed above and the long run population impacts associated with reduced survival and reproduction. Thus, they captured the fundamental dynamics of the fishery as well as the behavioral response of anglers. They estimated the welfare effects from several scenarios concerning the reduction of hypoxic conditions including improvements in water quality in only portion of the fishery vs. an improvement across the entire region. For a 25 percent increase in dissolved oxygen levels across the entire region, they estimated that catch rates would increase by about 20 percent, resulting in over $80 million dollar in annual benefits.

Lipton and Hicks (2003) study the effect of low dissolved oxygen on the striped bass fishery in the Chesapeake Bay and evaluated the welfare effects associated with the degrading

water quality in recreational fishing sites along the Patuxent River. While their results suggest that the welfare effects were relatively small due to the ability of anglers to easily substitute to recreation site elsewhere, they note that prolonged levels of anoxic conditions would result in annual angler losses of nearly $300,000, even for this relatively small area. As the authors note, these estimates do not include the dynamic effects that could impact reproduction and health of the population in the long run.

In a study focusing on the short run potential impacts of low oxygen on the blue crab fishery in three tributaries of the Chesapeake Bay, Mistiaen, Strand, and Lipton (2003) found evidence that these conditions lower harvest rates for a given amount of effort. Simulations suggest that a reduction in dissolved oxygen by about 30 percent would lead to a short run reduction in profits of over $225,000 for crabbers in the Patuxent River.

Less effort has been devoted to economic studies of the effects of hypoxia in the Gulf of Mexico, likely due to lack of clear evidence that important commercial and recreational fisheries have been severely impacted. However, Jenkins et al. evaluated the benefits of wetland restoration along the southern portion of the Mississippi River corridor. Among other benefits, these wetlands would help reduce the amount of nitrates entering the river and ultimately reaching the Gulf of Mexico. They estimate that the value of nitrogen mitigation from wetlands restoration in the region would fall between about $900 and $1900 per hectare of land considered, but these estimates are based on benefit transfer methods which may not accurately reflect damages in the Gulf.

Notably absent from the literature are studies that address lost nonuse or existence values associated with hypoxic conditions and eutrophication.

III. Policy Responses and Cost Effective Approaches to Abatement

In this section, we describe the policy responses, and the economic research related to two major areas of hypoxia (the Baltic Sea-largest in the world), and the Gulf of Mexico (second largest). We begin with a brief discussion of the policy setting surrounding both hypoxic areas and then discuss a number of common challenges that are relevant to the design and implementation of policy, including: the complex fate and transport of pollutants over large landscapes, nonpoint source pollution problems, multipollutant and multi-jurisdiction issues, bioeconomic modeling, temporal considerations, and uncertainty.

For the Gulf of Mexico, The 2001 Action Plan for Reducing, Mitigating, and Controlling Hypoxia in the Northern Gulf of Mexico (Mississippi River/Gulf of Mexico Nutrient Management Task Force 2001) called for reducing the size of the hypoxic area in the northern Gulf of Mexico to less than 5,000 km^2 over a five-year period by the year 2015, and for accomplishing these goals through voluntary actions, incentives, and education. In a more recent effort, the EPA Science Advisory Board (SAB) convened a Hypoxia Assessment Panel (HAP) that reviewed the science background, newer information on nutrient loads and sources, and a better detailed analysis of agricultural practices and economics within the Mississippi River watershed. Parallel to this effort, the federal-state Task Force began the formulation of a new Action Plan. The work of the SAB/HAP was completed in 2007, and the new Action Plan (Mississippi River/Gulf of Mexico Nutrient Management Task Force 2008) was completed. The SAB (2007) report supported and strengthened the science supporting nutrient management within the Mississippi River watershed to improve water quality both within the watershed and in the receiving waters of the northern Gulf of Mexico. The 2008 Action Plan maintains the 2001 goal of 5,000 km^2 by 2015, but drafters of the plan acknowledged that it might already be too

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late to reach that goal considering both the limited action to date and visible planned action in the near future. The revised Action Plan calls for states to devise solutions "as soon as possible, but no later than 2013."

The 2008 Action Plan calls for annual operation plans that outline measurable goals and an annual accounting of the actions toward the goals. The level of detail for the states and identifiable actions to meet the goals are disparate, uneven, and, in many cases, identified as "ongoing" or "continuing," with little evidence of proactive engagement in the implementation strategy development. By the end of 2012, the prospect for doing much more than providing some input to "revise this strategy" appears unrealistic.

In the meantime, Northern Europe has been grappling with its own eutrophication problem in the Baltic Sea for decades, and was somewhat ahead of the U.S. in terms of policy targets. The bulk of existing economics literature addressing marine eutrophication deals with the Baltic Sea region. The Baltic eutrophication problem has been on the European policy radar since the early 1970s, starting with the creation of the Helsinki Commission (Helcom) in 1974). Two rounds of ministry-level agreements on nutrient load targets occurred, in 1988 and in 2007. The 2007 plan, called the Helcom Baltic Sea Action Plan (BSAP), calls for the measures to be implemented by the Baltic coastline countries by 2021, and, unlike the U.S. hypoxia Action Plan, involves country-specific nutrient (nitrogen and phosphorus) abatement targets (Elofsson, 2010).

Overall, a significant body of both theoretical and applied economics research has focused on the issues pertinent to coastal eutrophication (and degradation of coastal ecosystems in general). Several important conceptual and methodological issues have been identified (Elofsson et al., 2003). The scale and scope of coastal eutrophication problems, where large drainage basins, often dominated by agriculture, industry, and urban areas, are often the main culprit, brings to

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the fore multiple issues in environmental economics research and certainly makes coordinated policy response challenging. Multiple pollutants, originating from a variety of sources (both point and non-point) in multiple political jurisdictions, and multiple possible abatement technologies, some of which require large capital investments and/or suffer from problems of monitoring and compliance, are involved. The transport and fate of multiple pollutants is difficult to characterize, requires significant data and modeling efforts, and is inherently stochastic (with some stochasticity endogenous to the choice of abatement technology).

The issues of eutrophication and its control are further dependent on concomitant pollution deposition from outside the watershed and on upstream impacts of pollutants. When a large area contributes to eutrophication, depending on the pollution control policy, general equilibrium effects may be important. Data needed for physical process modeling usually does not match well with the economic data on production or abatement activities. Despite these issues, a significant amount of important work has been done which can lay the theoretical foundation and provide the empirical strategies for future research.

We must note that the issue of pollution originating from non-point sources such as agriculture and urban areas is clearly central to the eutrophication issue both conceptually and in terms of relative impact of nonpoint sources. For example, Ribaudo et al. (2008) estimate that if point sources' emissions were fully eliminated in two-thirds of all nitrogen-impaired watersheds in the United States, the maximum achievable nitrogen reduction would be below 10%. Conceptually, there is a large body of both theoretical and empirical research devoted to nonpoint source pollution control (Shortle and Horan 2008; Xepapadeas, 2011; Russell and Clark, 2006 all provide excellent reviews). Most of the empirical applications in terms of nutrient reductions from nonpoint sources have focused on a relatively small scale, mostly due to the

extensive data requirements required to capture both the pollution processes and the behavior of individual economic agents. We believe, however, that advances in modeling and data availability are starting to make this literature very important for dealing with larger scale eutrophication problems.

In terms of economic studies focusing on the large regional, national, or international scale most relevant to marine eutrophication problems, the earlier studies include Johnsen (1993) and Paaby et al. (1996). The basic modeling approach is to cost-efficiently allocate abatement effort at the sectoral level (between agriculture and the wastewater treatment sector). Utilizing the same basic approach of looking at optimal allocation of abatement activities across sectors, Ing-Marie Gren and coauthors have produced an important body of work related to the entire Baltic Sea basin. In a first of a series of published papers, Gren et al. (1997) studied cost-efficient reductions in both nitrogen and phosphorus in order to reach the Helcom's 1993 target of 50 percent nutrient reductions to the Baltic Sea. The authors take the aggregated sectoral approach but model the full set of nutrient delivery pathways to the sea, including both water pathways (agriculture, urban sources) and atmospheric deposition from transportation and energy (combustion plants). With the Helcom target not clear as to how the 50 percent eventual abatement were to be achieved, the authors find very large cost savings in the case of optimally allocating nutrient abatement responsibility among the sectors in 9 affected countries, as opposed to the mandate of proportional reductions[2].

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[2] Since that time, several studies using a similar approach (reviewed in Elofsson, 2010) have been conducted for the Baltic. Cost-efficiency gains from optimal allocation of abatement effort across countries and industries are consistently large, yet a lot of effort is devoted to reiterating that point, which appears to be due to political considerations.

International and inter-jurisdictional dimensions

The case of the Baltic Sea, which is bordered by watersheds located in several quite heterogeneous countries in terms of their physical, economic, and political makeup highlights the issues of international cooperation, transboundary pollution, and managing the commons. Thus, the broader context (discussed, e.g., in Folmer and de Zeeuw, 2000) of international and inter-jurisdictional cooperation (or lack thereof) is applicable to many eutrophication problems which almost always involve multiple jurisdictions. In general, unilateral policies result in too much eutrophication. Interestingly, for the case of the Baltic Sea, where nitrogen and phosphorus act as limiting nutrients in different marine basins (at different times), each nation's focus on their limiting nutrient produces not only too much eutrophication, but too much of abatement of the "wrong" nutrient, due to the externality that the export of non-limiting nutrient has on the other country's basin (Turner et al., 1999, p. 343-344). For the Gulf of Mexico hypoxia, it is the interactions between upstream and downstream states and with the federal agencies that are most salient.

Interactions with other policies and environmental issues

Due to both the scale and the pollutant source and pathway nature of the eutrophication problem, interactions between other pollution problems and policies may become important. Burning of fossil fuels directly and indirectly affects the amount of nitrogen deposited to marine systems, and future climate change impacts affecting temperature, precipitation, and storm and current patterns are all likely going to affect the eutrophication problem. Thus, anthropogenic sources of climate change and associated policies may be important to consider. Several steps have been taken in terms of characterizing the likely impacts of climate change on riverine

nutrient loads to the Gulf of Mexico (Justic et al., 2003; Donner and Scavia, 2007). Overall, riverine delivery of nutrients is expected to increase with climate change.   Of course, a much more direct policy interaction is with agricultural policies, since agricultural nutrient contributions are often the most important source of nutrients. The share of agriculture's contribution varies depending on the particular system. In the Gulf of Mexico, agriculture contributes over 70 percent of total nitrogen and phosphorus (Alexander et al. 2008), while in the Baltic, agriculture was estimated to contribute about one-fifth of total nitrogen load and one-third of total phosphorus load (Turner et al., 1999). For the Baltic, the CAP reform has been found to have a beneficial impact on nutrient losses (Schmid et al., 2007), but sometimes the policy changes have the effect of shifting pollution from agriculture to other sources (Brady, 2003). For the U.S., federal conservation practice support, while not explicitly related to the Gulf of Mexico hypoxia reduction goal, has been estimated to reduce nutrient loadings significantly (USDA-NRCS, 2012).

Ribaudo et al. (2005) investigates the agricultural market equilibrium impacts of a nitrogen credit trading program, where point sources would be subject to strict nitrogen abatement technology standards but would be allowed to purchase nitrogen credits from agriculture. Allowing farmers in affected regions to optimize acreage decisions taking into account the price of nitrogen credits, the authors find that there are some market spillover impacts, with the planted acreage increasing both within and outside the nitrogen trading region (by 0.8 percent overall). Interestingly, Ribaudo et al. (2005) find that agriculture ends up undertaking 91 percent of nitrogen abatement effort.

In addition to causing eutrophication problems, nutrients often impair upstream rivers, lakes, and streams. Upstream policy responses to deal with local pollution are generally not coordinated

with any effort regarding downstream eutrophication effects. For example, in dealing with nitrate pollution of drinking water, the City of Des Moines in Iowa, USA, operates the world's largest denitrification facility. Once nitrogen has been removed from the drinking water, it is released further downstream. Similarly, one needs to be cognizant of the upstream benefits of nutrient load reductions in evaluating marine eutrophication policies.

Need for bioeconomic modeling on land and sea

Large-scale studies which incorporate economic heterogeneity across countries, sectors, and abatement technologies typically either lack the spatial component or operate on the scale of a country (Gren 2001) or a similarly large region (Ribaudo et al., 2005). For water-borne nutrients, especially originating from agriculture, pollution fate and transport is critical, and the relevant spatial unit is that of a watershed. The level of watershed resolution (since an infinite number of watersheds can be defined for any unit of land area) and the level of resolution of various polluting entities within the watershed are fundamentally important for policy analysis, choice of economic instruments, and "targeting".

Essentially, bioeconomic modeling on land becomes a key component of dealing with the eutrophication problem (the need for bio-economic modeling in marine environments is obvious if the hypoxia impacts described above are to be appropriately considered). Agricultural and environmental economists have long incorporated biophysical models into economic modeling and policy analysis for water quality (e.g., Braden et al., 1989 in the context of sediment control)[3], but it is only fairly recently that advances in modeling and computational capacity have allowed for their use in larger-scale studies appropriate for evaluating policies aimed at reducing nutrient delivery to coastal ecosystems (e.g, Kling et al., 2006). Of particular interest to

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[3] And many other contexts such as air quality, water quantity, habitat preservation, forest ecosystems, and global climate change. Polasky and Segerson (2009) discuss the promise and the challenges of integrating ecological and economic models.

economists are process models which can simulate environmental impacts of decisions on the scale of economic agents (e.g., farms, municipalities, etc.).[4] Another attractive feature of many process models is their ability to explicitly operate on a much finer (and thus potentially much more policy-relevant) spatial scale and to provide the policymakers and stakeholders other relevant information (local pollution impacts, information on other pollutants) in the same framework. Recently, Rabotyagov et al. (2010) focused on agriculture in the Upper Mississippi River basin and used spatial multiobjective optimization heuristics to allocate abatement effort across subwatersheds to achieve significant nutrient reductions. Their model considers fertilizer reductions, agricultural conservation practices and land retirement as nutrient abatement technology and does not include the ability to fully represent agricultural market interactions (as in Ribaudo et al., 2005) or other relevant technology such as wetlands (e.g., Gren et al. 1997). However, it does provide for a more spatially explicit analysis and scientifically accurate representation of the fate and transport of nutrients.

In terms of the need for better bioeconomic analysis of eutrophication problems at sea, the spatial dynamics of eutrophication in the Baltic Sea provide a clear illustration, where nutrients deposited at one location move across various marine basins and contribute to eutrophication problems (Savchuck and Wulff, 2009). Gren (2009) uses this information to better allocate abatement effort in the Baltic drainage basin. The impacts of eutrophication on the dynamics and the composition of economically important marine organisms clearly demand methods described in Smith et al. (2009) and begin to be elucidated by studies like Knowler et al. (2001) for the Black Sea and Huang and Nichols et al. (2012) for North Carolina.

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[4] Ribaudo et al. (2005), among others, use biophysical models to estimate nutrient impacts of economic policies, but they were limited to edge-of-field models which do not accurately describe pollutant loadings at relevant spatial scales (e.g., watershed outlet).

Temporal dimension

In considering the processes that characterize eutrophication, it also becomes clear that time can be an important dimension to consider. For the Baltic, the impact of nutrients, especially phosphorus, can be felt for decades (Turner et al., 1999, Gren, 2009). Several studies deal with temporal effects of eutrophication at different scales. Gren (2009) considers a 60-year horizon to look for dynamic spatially efficient allocation of nutrient abatement effort. Goetz and Zilberman (2000) develop a theoretical model of spatial dynamic nutrient control, where the problem is split into two stages, and the spatial solution is optimized over time. Hart (2003) models lengthy time lags which may occur due to upstream nitrogen accumulating in groundwater and considers optimal allocation of upstream (agricultural) versus downstream (mussel cultivation) abatement. The question of irreversibility, either in terms of abatement investment (e.g., investment in municipal wastewater treatment plants, modeled by Laukkanen and Huhtala, 2008) or in terms of ecosystem response thresholds (Naevdal, 2001; Mäler et al., 2003) has been investigated in, with results being highly sensitive to eutrophication process assumptions. The importance of time lags, whether the eutrophication problem should be approached from the stock pollution problem perspective, and whether thresholds are present all appear to be an empirical question (to which we make a small contribution below).

Uncertainty

Uncertainty is endemic to all water pollution processes, and nonpoint source emissions (and abatement) are inherently stochastic. In the nonpoint source literature and policy, it has often been assumed that point source abatement is much less uncertain compared to the nonpoint-source abatement, which has led to preference toward point source abatement. As Shortle and Horan (2008) discuss in their review, this does not always follow, and a social planner may wish

to target more variable nonpoint sources precisely in order to reduce overall pollution damage uncertainty. Conceptually, when different abatement measures are uncertain and distributions can be quantified, probabilistic constraints can be specified and transformed into deterministic equivalents amenable to optimization by methods like the chance-constrained programming (Charnes and Cooper, 1963). This approach has been fruitfully pursued for many water pollution problems, and has been adopted for large-scale studies focusing on eutrophication in works by Byström et al. (2000) and Gren et al. (2002). Recently, Gren (2010) used this approach to quantify the resilience value of constructed wetlands, and Rabotyagov et al. (2010) use a similar approach to find spatial combinations of nutrient-reducing practices which are resilient with respect to weather shocks. One important consideration is that these kinds of approaches are capable of dealing with quantifiable uncertainty (risk) and not the epistemological uncertainty stemming from fundamental lack of understanding of ecosystem processes.

IV. Linking the size of the zone to its inputs: the Gulf of Mexico Hypoxic Zone production function

In this section, we aim to reduce some of the knowledge gaps regarding the basic physical linkage between pollutants and hypoxia in the Gulf of Mexico. Ecological theory suggests that the extent of eutrophication should be a function of nutrient inputs. The SAB report reiterated that nitrogen loading drives the timing and extent of hypoxia. The SAB further included phosphorus as a nutrient that needed management. Thus, the Action plan recommends a dual nitrogen and phosphorus reduction strategy. Given that the most current analysis of sources of nutrient loads from the Mississippi River watershed indicate that agricultural sources in the watershed (Figure 3) contribute more than 70 percent of the delivered nitrogen and phosphorus (Alexander et al., 2008), and that tile-drained, corn-soybean landscapes are very nitrogen 'leaky',

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the scientific consensus is that nutrient abatement should target the tile-drained Corn Belt region of the Mississippi River watershed for nitrogen and phosphorus reductions in both surface and sub-surface waters.

Despite the strong policy focus and well-established theoretical importance of nutrients in limiting phytoplankton growth, statistical evidence regarding the role of multiple nutrients in contributing to the size of the hypoxic zone has been sparse. In fact, debate on the factors responsible for the severity of the hypoxic zone goes back as far as Wiseman et al. (1997), who first observed a relationship between Mississippi streamflow and the area of hypoxia. Wiseman et. al and other efforts have been made possible by the time series of systematic hypoxia observations going back to 1985 (Rabalais, Turner et al., 2007; Rabalais, 2011). In this section, we briefly review the existing work and contribute to the debate by estimating a new empirical relationship for the hypoxic zone where we find evidence that all macronutrients (Nitrogen (N), Phosphorous(P), and Silicate (Si)) explain the extent of hypoxia and that lags in the impacts of nutrients are important. These "ecosystem memory" relationships (Turner et al., 2006, others) have long been postulated but until now have not been statistically corroborated.

Several studies have made use of the Streeter-Phelps model, which is typically used to predict the impacts of point source loads on oxygen concentrations downstream.. Examples include Scavia et al. (2003), Scavia and Donnelly (2007), and Liu et al. (2010). Calibration of parameters of the model by year has produced good fit to the observed hypoxia area (88 percent of variability explained in Scavia et al. (2003), and 99 percent explained in Liu et al. (2010) using Bayesian calibration techniques. All of these models focus on nitrogen. Nitrogen, however, is not the only limiting factor in phytoplankton growth, and, while in steady-state, marine environments are P-limited (Tyrell, 1999), the dynamics of the coastal systems result in

some portions of the zone being N limited while other are P limited (Quigg et al. ,2011; Turner and Rabalais, 2012).. As a result, a simple Leontief-type production function is not likely to accurately describe the underlying production process for the average size of the overall zone. If the process were of simple Leontief type, abatement focus should be on a single nutrient (Kuosmanen and Laukkanen, 2011). However, when areas of different input limitation are present, the observable "hypoxia production process" will appear to exhibit substitution between N, P, Si, and other limiting inputs (e.g., light). Consider a simple example, where the overall hypoxic area consists of two sub-areas, one of which is N-limited while the other is P-limited. Suppose increased N load is delivered. The N-limited area will increase, while the P-limited area will not. Now suppose P load is increased. The converse will be true (P-limited area grows, N-limited area does not). Again, this observationally will appear as substitution among inputs. Furthermore, in regions where areas of different nutrient limitation can exchange surplus nutrients over time, abatement of a single nutrient will free up the excess input to contribute to another area's production. This has important management implications, including the possibility of inefficient outcomes when different jurisdictions impose an externality on each other by focusing on a single nutrient only (Turner et al., 1999).

While the scientific community is well aware of these issues and, as a result, the current Action plan calls for a reduction in phosphorus in addition to nitrogen, the empirical corroboration of the importance of multiple nutrients has been elusive. Several factors appear to have contributed. First, the process of hypoxic zone formation is a complex, dynamic, and heterogeneous, potentially subject to exogenous impacts of winds, ocean currents, and disruptive weather events (storms and hurricanes). Second, the history of nutrient enrichment of the shelf may have confounding effects. Turner et al. (2006) speak of "ecosystem memory", where system

attributes are carried from one year to the next, "as would happen if organic matter accumulating in one year was metabolized over several years." This suggests the potential importance of lagged inputs in the hypoxia production function. Finally, the system may exhibit transitions to different states (Kemp et al, 2009 Rabalais et al., 2010), hysteresis and, empirically, the existence of time trends and possibly serial correlation, even after lagged inputs are accounted for.

Previous statistical models include Turner et al. (2006), who postulated that nitrogen , phosphorus, silicate, as well as lags in nutrient inputs as short as 1 month before the hypoxia size measurement in July and as long as 3 years prior would be significant in explaining hypoxic area variability. They found that nitrogen loads, as well as phosphorus loads, lagged one or two months before July were significant. However, the best model in their study was one relating May nitrogen and a time trend to the size of hypoxia (explaining 82 percent of observed variation).

Greene et al. (2009) used a stepwise regression (both backward and forward selection) procedure to choose the month of nitrogen and phosphorus concentrations and river flows to be included in the model. In the best models reported, May nitrogen concentrations, May flows, and February phosphorus concentrations were reported as significant. They also report that a post-1993 dummy variable was significant, and interpret potential evidence of a regime shift in the system (and the inclusion of the dummy made the impact of phosphorus non-significant).

Finally, Forrest et al. (2012) explored the power set of 11 regressors using Bayesian Information Criterion for model selection. Their primary focus was on identifying the impacts of nitrogen concentrations, a time trend (or a changepoint) and wind patterns (postulated to affect hypoxia formation). They, too, found evidence of positive impact of nitrogen, of a linear time

trend, or a post-1993 regime shift, and a negative impact of the eastern component of the wind on hypoxic zone size. The role of wind forcing was also found by Feng et al (2012), where the duration of westerly wind was also found to negatively affect the size of hypoxia.

We build upon the existing work and estimate the area of the hypoxic zone using simple production function estimation methods and attempt to account for the possibility of multi-year lags, importance of multiple nutrients, and the time series structure of the data. We estimate a translog production function in three inputs (nitrogen, phosphorus, silicate), allowing for lags in responses and for serially correlated errors. Specifically, we estimate:

$$HypoxicZone_t = \beta_{intercept} + \beta_{hurricane}Hurricane_t + \beta_{current}Current_t + \beta_{hurrN}Hurricane_t * N_t$$

$$+ \beta_{hurrP}Hurricane_t * P_t + \sum_{i=0}^{5} \beta_{i,N}N_{t-i}$$

$$+ \sum_{i=0}^{5} \beta_{i,P} P_{t-i} + \beta_{Si}S_t + \beta_{windpower}Windpower_t + \beta_{SeaTemp}TempAnomaly_t + \varepsilon_t$$

As postulated in much of the existing literature, the areal extent of hypoxia (in km^2) depends oncurrent and past loadings of nitrogen (N_{t-i}, as log$_{10}$ transform of May USGS estimates of mainstem Mississippi River loads), phosphorus (P_{t-i}, log$_{10}$ transform of June USGS estimates), and contemporaneous silicate (S_t, log$_{10}$ transform of May USGS estimates).[5] We explore the effect of hurricanes and unusual current conditions and their interactions with nitrogen and phosphorus. We use Forrest et al. (2012) data on observed "wind power" and the Sea Surface Temperature anomalies ($TempAnomaly_t$) in the regression to capture local meteorological factors. We estimate the model using hypoxia size measurements from 1985 to 2008 (excluding

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[5] USGS Nutrient loadings at St. Francisville, available at:
http://toxics.usgs.gov/hypoxia/mississippi/flux_ests/delivery/index.html, document Gulf-Monthly-2011.xls).

1989 when lack of funding precluded data collection), and use the years of 2009 and 2010 for prediction to assess model performance. Estimating separate coefficients for each lagged nutrient values using the available 23 observations would quickly use up degrees of freedom, so a polynomial distributed lag model (Greene, 2003, pp. 555-556), which constrains the lagged regressors' parameters to lie on a polynomial, was used. While higher-degree polynomials were tested, a linear form of the lag distribution was used. Since the hypoxia area data are collected over time, an autoregressive error model was estimated. That is, the error term is assumed to follow $\varepsilon_t = u_t - \theta \varepsilon_{t-1}$, where $u_t \sim N(0, \sigma^2)$.[6] Table 1 presents the results. The model suggests that spring nutrient (N, P, Si) loadings and other factors (hurricanes, currents, wind conditions, and sea temperature anomalies) affect the areal extent of the hypoxic zone. While all nutrients are found to positively affect the size of hypoxia, both nitrogen and phosphorus have a persistent (and diminishing over time) impact on hypoxia, with the impact being significant up to 5 years into the future. This is consistent with previously postulated "ecosystem memory" conjecture, although the time dimension was found to be less important than what is suggested for the Baltic Sea, as riverine nutrient loadings older than 5 years were not found to significantly impact the size of hypoxia.[7] Clearly, this may have important implications for management and for the choice of whether to pursue dynamic approaches in economic and policy work.

Previously, it was suggested that there may have been a threshold effect, or a "regime shift" for the Gulf hypoxia. In particular, it was suggested that the system response changed post-1993 and Turner et al. (2012) found that in a model of hypoxia response to nitrogen, there is a significant time trend. In our model, once the lagged nutrient impacts were accounted for, we

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[6] Longer autoregressive lags in the error term were explored as well but no significant evidence for them was found. In addition, we tested and did not find significant evidence of heteroskedasticity.

[7] Interestingly, in the Black Sea natural experiment, as fertilizer used declined rapidly, the hypoxic area persisted for about 5 years (Kemp et al., 2009).

could not find significant evidence of a threshold effect of a post-1993 era or evidence of a time trend.

We do corroborate the importance of factors other than nutrients, controlling for effects of hurricanes, wind power, currents, and surface temperature anomalies. Hurricanes are a major disturbance and may affect current and future hypoxia formation in complicated ways . While hurricanes, by mixing the water column, reduce the size of current hypoxia, we also find that the impact of current year's nutrients is modified by the presence of a hurricane. In particular, the impact of contemporaneous nitrogen is amplified ($\beta_{hurrN} > 0$), while the impact of contemporaneous phosphorus is mitigated ($\beta_{hurrP} < 0$). [8] The presence of unusual currents has a negative impact on the size of the hypoxic zone. Given that the impact of wind has been posited and explored (Forrest et al. 2012) and Feng et al. (2012), we tested wind variables defined and discussed in Forrest et al. (2012). Their "wind power magnitude" variable, defined as the magnitude of the southwesterly wind vector (geometric mean of the eastward and northward components of the wind), was found to be significant. We find a positive relationship between hypoxia size and wind power. Winds can potentially shift the river plume and shift the zone where hypoxia may develop (Forrest et al., 2012). As suggested by Kemp et al. (2009), "summer winds are also important in regulating NGOM hypoxia, where up-coast (southwesterly) winds favor retention of freshwater (plus nutrients and plankton) on the Louisiana-Texas shelf and enhance stratification and hypoxia[…]". Our results support this theoretical conjecture.

Sea temperature anomaly, which was first explored by Forrest et al. (2012) was found to be significant at the 10 percent level of significance, and further suggests the importance of

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[8] We also estimated a model for the period prior to 2003 (no recorded hurricanes). Findings with respect to impact of nutrients are very similar.

weather and that the changing climate may have a strong influence on the size of the hypoxic zone.

We leave 2009-2010 observations to assess how well the model predicts the hypoxic area.[9] As Figure 4 shows, the model performs fairly well, predicting the areal extent almost exactly for 2010, but underpredicting the 2009 zone by about 6,000 km^2.[10]

Implications for policy goals

The policy goal for the Gulf hypoxia is 5,000 km^2 or less, computed as a five-year average. At the time of the 2001 Action Plan and consistent with historical data and model predictions, it was predicted that a 30 percent nitrogen load reduction was likely needed to reach the 5,000 km^2 environmental goal. Subsequently, given the increased size of the hypoxic zone and the increase in nitrogen load per unit volume of water, it was predicted that nitrogen load reductions closer to 35 to 45 percent would be necessary to reduce the size of the midsummer hypoxia (Scavia et al. 2003, Justić et al. 2003). Using the estimated model, we can consider the counterfactual of reducing the observed nitrogen and phosphorus loads and the implications for reaching the Action Plan goal (Figure 5).

We estimate that a concomitant 30 percent reduction in May nitrogen and June phosphorus loads would have been sufficient to reduce the areal extent of hypoxia below the 5000 km^2 goal (and a joint 40 percent reduction in nitrogen and phosphorus puts the upper bound of the 95-percent confidence interval of model prediction below the policy goal).

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[9] 2011 observation is not included since the weather variables computed by Forrest et al. (2011) only extend through 2010.

[10] Researchers mapping the hypoxic zone noted that "During the summers of 1998 and 2009 several weeks prior to the mapping cruises, the winds were stronger than usual from the southwest. This resulted in water masses being pushed towards the east. Thus hypoxia that may have been a feature of the upper Texas coast or the western Louisiana shore were piled up to the central and eastern coasts of Louisiana. Wind measurements and current measurements confirm this phenomenon. As a result the "footprint" of bottom-water hypoxia was smaller, but the total volume was consistent with other years. i.e., the hypoxia was thicker on the central and eastern portions of the area mapped" (http://www.gulfhypoxia.net/Research/Shelfwide%20Cruises/).

Consistent with the recent scientific consensus, we do find that a 30 percent in nitrogen alone would not have been sufficient to reach the goal. Of course, while these results may inform the translation of the hypoxia zone goal to sufficient nutrient reduction goals (at least within the historic range of variability), we should note that the proportional reduction in nitrogen and phosphorus is not likely to be the cost-effective allocation of nutrient reductions. Instead, the estimated "hypoxia production function" should be used in selecting the minimum-cost combination of inputs (nutrients) which achieve the policy goal. A two-nutrient abatement cost function, akin to one estimated by Rabotyagov et al. (2010) for the Upper Mississippi River watershed, can be used to find the efficient mix of nutrient targets (and the corresponding spatial allocation of abatement effort across subwatersheds and abatement measures). At the very least, the estimated hypoxia production function can provide for a better assessment of nutrient reduction policies and proposals.

V. Closing Thoughts

The problems of eutrophication and their associated "dead zones" are globally occurring; increasing in both size and duration at an alarming rate. There are common characteristics of many of the affected areas, but differences in the physical, economic, and policy dimensions mean that the lost ecosystem services and most cost-effective abatement approaches will need to be assessed on a case-by-case basis. Nonetheless, designing effective policies to internalize the costs of these externalities requires addressing a series of common economic challenges including improved integration between ecologic and economic models, explicit recognition of spatial interdependencies, attention to the dynamics of pollution and economic processes, the treatment of uncertainty, policy interactions, and a range of political economy considerations.

Eutrophication problems highlight the need to improve our understanding of the range of ecosystem services and potential welfare losses and gains from policy action. This in turn, requires better understanding of the linkages between the sources of nutrients in the watershed, the fate and transport of those nutrients as they move through the watershed, and their impacts on the physical ecosystem services of aquatic systems. Economists need to work closely with ecologists and other natural scientists so that their findings can best be used in integrated assessments. One specific area where such integrated work is apparent is the need for more bioeconomic modeling in terrestrial ecosystems, including agricultural and urban areas. This work is needed to augment (certainly not replace) continued bioeconomic modeling of aquatic ecosystems. One area where additional bioeconomic modeling of aquatic ecosystems is needed relates to the consequences of nutrient over-enrichment on species that are not commercially important, but are instead aesthetically enjoyable, provide significant existence values, or are key to overall ecosystem stability (Barbier, 2012).

As a step towards better understanding the linkages between nutrient enrichment and dead zones, we present estimates from an ecologically-informed hypoxia production function for the Gulf of Mexico. We demonstrate the potential policy value of the production function by using it to assess the nutrient reductions that might be needed to meet alternative targets for the average extent of the hypoxic zone. While far from the last word on the empirical relationship between nutrients and the size of the dead zone, our findings provide new insight into the contribution of multiple nutrients and importance of time lags.

We close with an observation about another set of linkages which need to be better developed before sustainable solutions can be found: the linkage between the findings from integrated economic and ecological models and policy decisions. Too often it seems that careful

analysis is undertaken that provides useful insight into the design of policy that is not incorporated. For example, despite the fact that studies of the Baltic Sea reaffirm the efficiency gains from targeting, it appears that little serious attention in the policy process has been given to developing clean-up plans that target the least costly abatement locations and methods. Given the enormity of the challenge, approaches that rely on equal abatement effort will almost certainly involve much higher cost and ultimately result in delays in managing the problem. Inefficient policies to implement abatement actions (such as the equal sharing of the burden irrespective of cost-effectiveness) are ultimately a drain on the sustainability of a system. Economists need to do a better job of communicating that the efficiency properties they study contribute to the well-being of humans and the environment.

Acknowledgments

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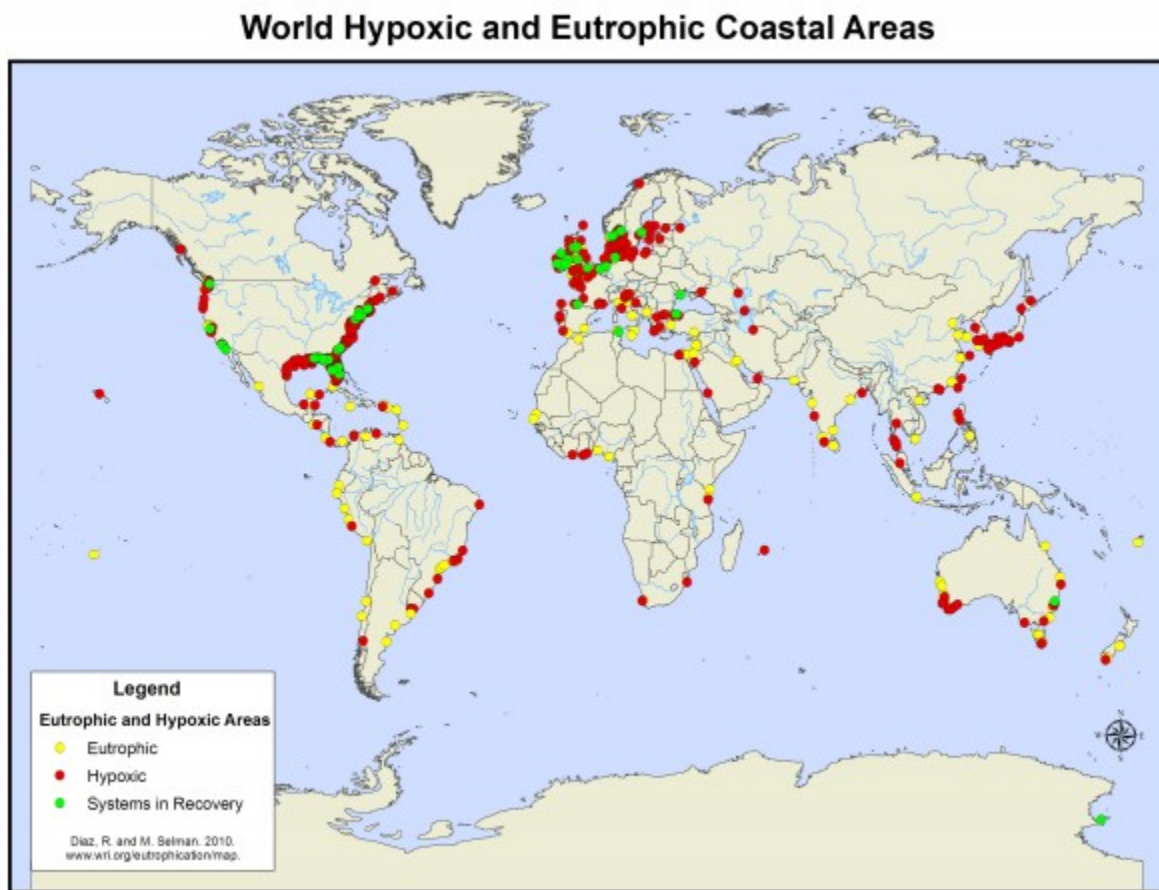
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Figure 1: Location and Size of Worldwide Hypoxic Zones[11]

World Hypoxic and Eutrophic Coastal Areas

Figure 2. Frequency of mid-summer bottom-water hypoxia (≤2 mg O_2 l^{-1}) off the coast of Louisiana and Texas for 60 to 80 stations (small dots) sampled during the summer from 1985 to 2008.
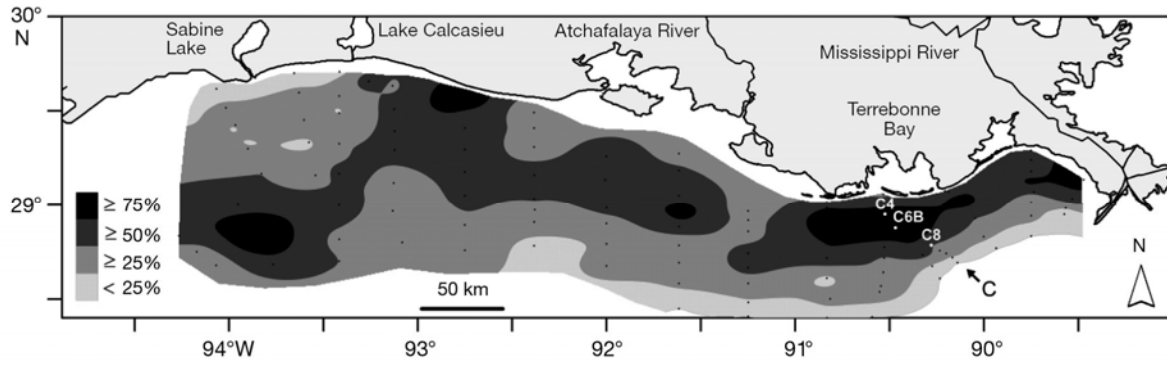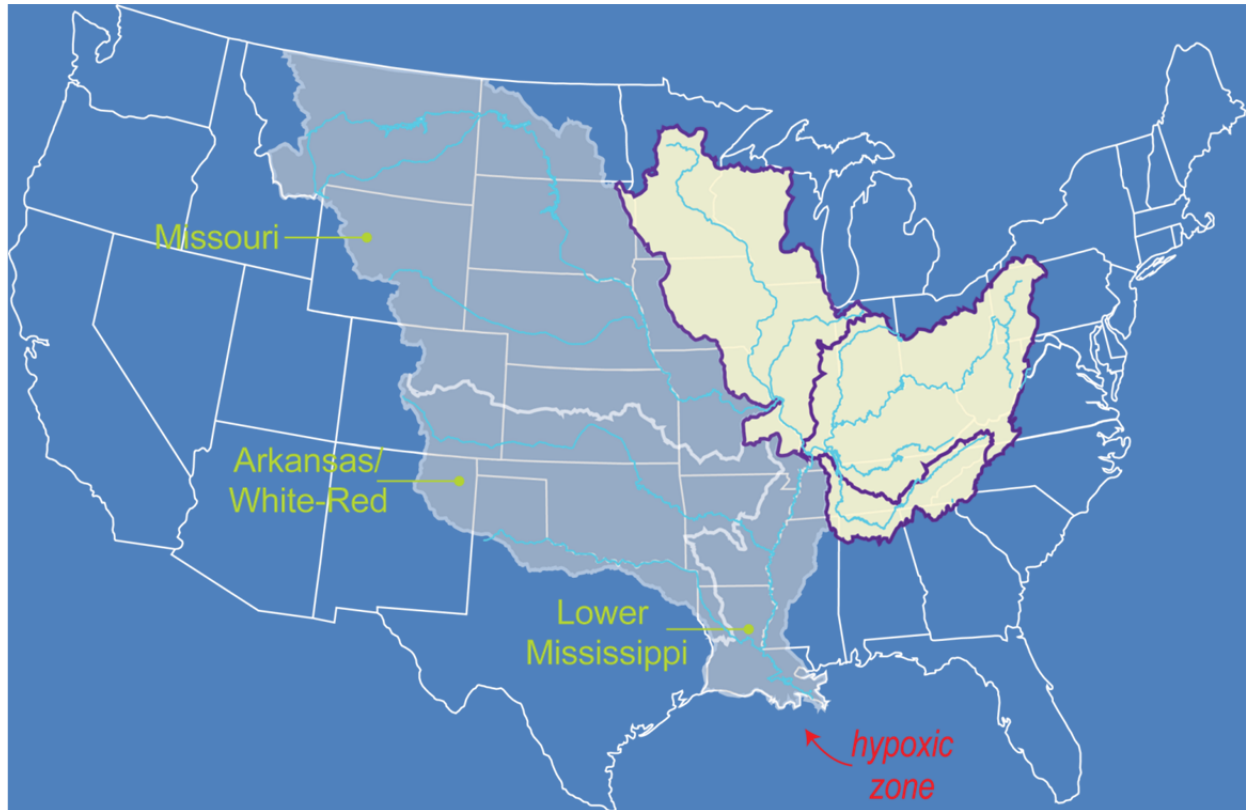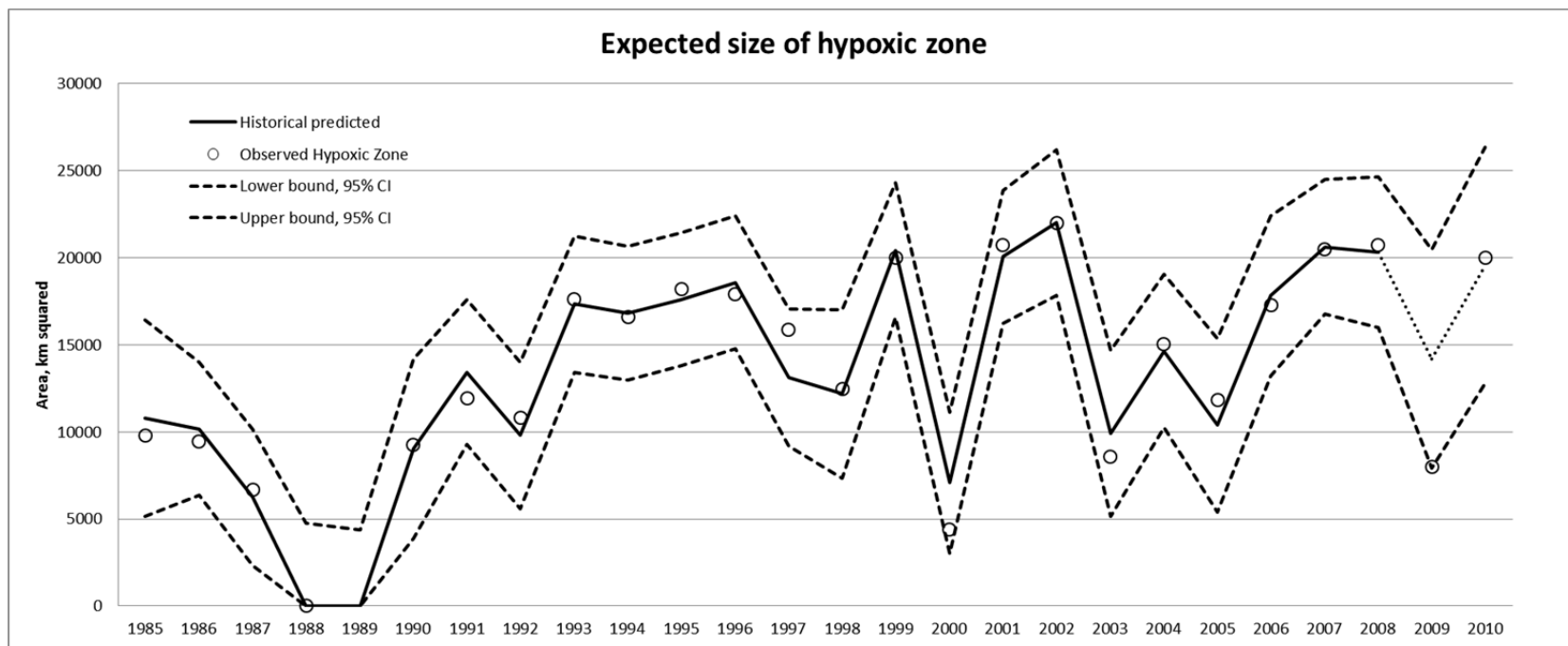
Figure 3: Mississippi River Basin and the Location of Gulf of Mexico Seasonal Hypoxic Zone
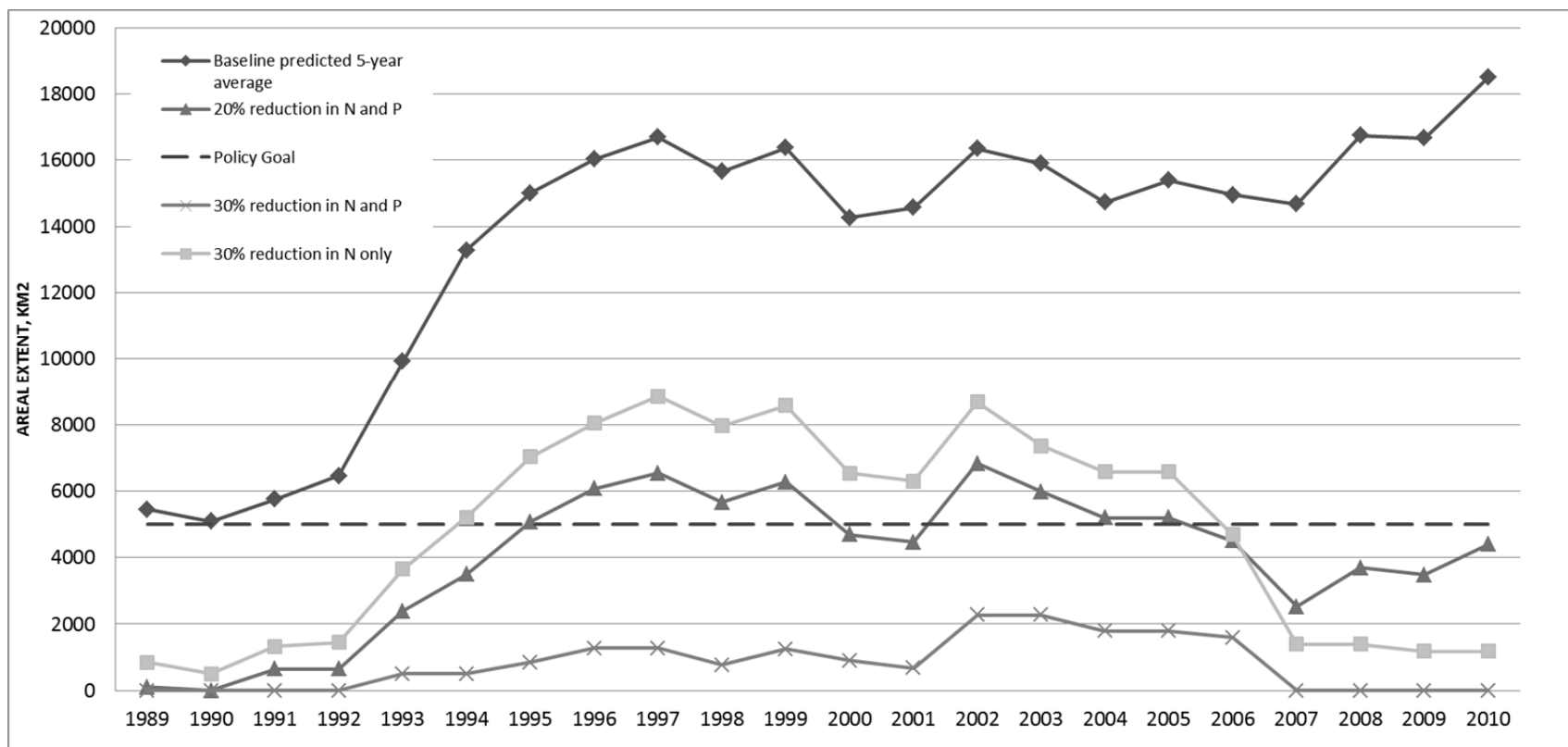
1   Figure 4. Observed and estimated area of Gulf of Mexico hypoxia (2009 and 2010 used for prediction)

2

3

4   Figure 5. Observed and counterfactual scenarios of 5-year average extent of Gulf of Mexico hypoxia

5

6

7

8   Table 1. Coefficient estimates for the Gulf of Mexico hypoxia "production function"
9

Parameter	Description	Estimate	s.e.	t Value	p-value
$\beta_{intercept}$	Intercept	-515666	37342	-13.81	<.0001
$\beta_{hurricane}$	Hurricane dummy	-321514	38944	-8.26	<.0001
$\beta_{current}$	Currents dummy	-5653	1925	-2.94	0.0149
β_{hurrN}	Hurricane-N interaction	120202	11690	10.28	<.0001
β_{hurrP}	Hurricane-P interaction	-69618	7723	-9.01	<.0001
$\beta_{0,N}$	N load, current year	14702	2997	4.91	0.0006
$\beta_{1,N}$	N load, 1-year lag	12262	2409	5.09	0.0005
$\beta_{2,N}$	N load, 2-year lag	9822.70	1869	5.26	0.0004
$\beta_{3,N}$	N load, 3-year lag	7383.30	1429	5.17	0.0004
$\beta_{4,N}$	N load, 4-year lag	4943.90	1206	4.10	0.0022
$\beta_{5,N}$	N load, 5-year lag	2504.50	1315	1.90	0.0861
$\beta_{0,P}$	P load, current year	19796	2175	9.10	<.0001
$\beta_{1,P}$	P load, 1-year lag	15523	1525	10.18	<.0001
$\beta_{2,P}$	P load, 2-year lag	11249	1122	10.02	<.0001
$\beta_{3,P}$	P load, 3-year lag	6975.86	1238	5.63	0.0002
$\beta_{4,P}$	P load, 4-year lag	2702.38	1773	1.52	0.1585
$\beta_{5,P}$	P load, 5-year lag	-1571.09	2469	-0.64	0.5389
β_{Si}	Si load, current year	9435	3534	2.67	0.0235
$\beta_{windpower}$	Wind Power	35.04	6.16	5.69	0.0002
$\beta_{SeaTemp}$	Sea Surface Temperature Anomaly	-3766	1922	-1.96	0.0786
θ	Autoregressive order-1 parameter	0.74	0.21	3.53	0.0055

Number of observations   23
Degrees of freedom, error 10
Total R^2 0.9662
Log-likelihood -193.28185
Bayesian Information Criterion 427.33

10